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Macrobenthos Responses to Dam Removal and Habitat Restoration in the West River, Connecticut

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MACROBENTHOS RESPONSES TO DAM REMOVAL AND HABITAT RESTORATION IN THE WEST RIVER, CONNECTICUT

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ABSRACT:

Dam removal is being increasingly used nationwide to restore impaired rivers and streams. While dam removals are becoming more prevalent, little is known about whether these efforts create conditions for the enhancement and or establishment of desired biota. Ongoing monitoring of these projects is an important step in the restoration process to ensure project goals are being met and the restoration has been successful. This thesis focuses on the Pond Lily Dam removal and restoration which took place in October 2015 along the West River in New Haven Connecticut that removed an aging mill dam with the objective of restoring the impoundment area back to a more natural habitat. Macrobenthic invertebrates were collected at the restoration site to perform an ecological assessment of the efficacy of the restoration in comparison to the existing Konolds Pond dam and impoundment as a control. Using multivariate analysis, community structure was analyzed to track the response of the benthic community at the Pond Lily Dam site and understand if the restoration had a positive impact on the ecosystem during the year following restoration. Based on the community composition, and diversity, the Pond Lily Dam ecosystem responded positively to restoration the year following removal of the dam. However, community composition was still highly variable and no apparent climax community had been reached. Based on this research, it is suggested that monitoring continue to better understand how benthic communities respond to major restoration efforts and to ensure the Pond Lily Dam site continues to improve and provide high quality habitat for native species.

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Chapter I
Introduction:

Across the country, dams have been built to suit a wide variety of societal needs such as energy generation, creation of lakes/impoundments for drinking water resources and recreation. Dams can range in size from only a few feet in height to those large enough to create large reservoirs for cities. It is well known that the construction of dams interrupts the natural hydrology of lotic ecosystems by disrupting the natural flow of water, sediments, nutrients, and organisms in river systems (Grill et al. 2015). Nationwide, many of the streams and rivers that were dammed have been targeted for removal on the local, state, and federal level due to disrepair or lack of continued usefulness (Millens and Wanstreet 2010). Such river reclamation projects are increasing in frequency in order to restore waterways back to more natural states for the recolonization of the natural ecological communities that existed prior to damming (Pollard and Reed 2004, Maloney et al. 2008). Macrobenthic invertebrates are especially sensitive to changes in the physical and chemical conditions of a stream, and changes in these factors can quickly alter benthic community composition (Doyle et al. 2005).

Damming of rivers and the creation of a reservoir can greatly effect abiotic factors within the river ecosystem by interrupting natural flow conditions in the river. Areas downstream of dams see decreased turbidity due to sediment retention within the impoundment; which in turn cause decreases in nutrients able to be transported downstream. Fantin-Cruz et al. (2015) showed this effect in a before- and after- dam construction experiment which showed a 38% decrease in turbidity, 23% decrease in total solids, 28% decrease in phosphorus, and 14% decrease in nitrogen after a “run-of-river” dam was installed on a river in Brazil. This suggests that river impoundment creates a nutrient and sediment sink in the impoundment.

Flow regimes in river ecosystems play a vital role in defining the distribution and abundance of biota (Schlosser 1985, Jones et al 2014, McManamay et al. 2015) and have a significant influence on changes in water chemistry, river morphology and community composition (Poff et al. 1997). Dewson et al. (2007) discussed the influence of low flow events on macroinvertebrates and found there to be significant responses by macrobenthos to stream flow, but inconsistent response trajectories. They found that in most cases, decreased flow rates caused a decrease in invertebrate density and taxonomic richness. Decreased density was often attributed to a reduction of space and increased competition within the reduced stream while reduced taxonomic richness was often attributed to a reduction of habitat variability.

Dams rapidly change the natural hydrology of rivers by reducing the magnitude and frequency of high water events. Natural variation in stream flow volume provides the natural dynamism in river channels through periods of low and high flow as well as extreme incidents of drought and flood. High flow periods are important for the transportation of sediment and nutrients downstream while suspended sediments can scour substrates allowing for communities to maintain their natural dynamic equilibrium (Grill et al 2015). During rare extreme flood events when the river extends outside of the channel and into the floodplain, nutrients are deposited into the riparian zone while woody and other large debris can be transported into the channel to create new habitat while maintaining higher biodiversity in the river and surrounding areas.

Low flow events also provide recruitment opportunities for species specialized to such harsh conditions. Extreme low flow events cause a significant loss in aquatic species, as space becomes the limiting factor. During these events, stream eddies, pools, and backwaters become areas of refuge for native species adapted to low flow conditions. By stabilizing flow regime, invasive and non-native species adapted to steady flow conditions are not exterminated and can proliferate in areas without extremes in river flow (Brooks 2016).

Although water quality is affected through the change in flow regime in a river, geomorphological effects of reduced sediment loads will influence downstream characteristics of the river bed. Walther (2016) showed there was a significant increase in suspended grain-size downstream of a dam on the McKenzey River in Oregon with more than a 50% increase in medium and large grain-sizes after the dam was constructed Walther (2010) also found that flow stabilization had significantly changed the flow regime of the river, raising mean low flows, and lowering high flow events which in turn has reduced the geomorphic complexity across the research area.

Studies on the effect of impoundments on benthic communities have shown wide ranging impacts. In 2015, Mbaka and Mwaniki showed inconsistent effects of impoundments across varying sizes of impoundments. Most studies showed no that impoundments had no significant downstream effects on physio-chemical characteristics. Community metrics, such as invertebrate diversity and abundance appeared to decrease around low-head dams (5-15m), but generally show no effect, or variable positive and negative effects around small dams (<5m) and run-of-the-river dams. In the studies focusing on run-of-the-river dams, a wide range in effect size correlations on abundance and diversity (-0.77-0.65) suggest great variability in communities downstream of stream impoundments. Community variability downstream of impoundments make restoration efforts difficult to understand, especially without a thorough understanding of the community structure prior to restoration efforts.

To rehabilitate streams, many efforts employ active restoration techniques where gravel, rocks, or large woody structures are added to river beds to encourage the reestablishment of natural faunal communities (Harrison et al 2004). These methods have been successful in reestablishing more natural stream morphology and hydrology, but there is little evidence to show that they also encourage recolonization by pre-disturbance biota (Bernhardt and Palmer 2011). While restoration attempts have been on the rise across the globe, evidence of ecological benefits from restoration are limited (Friberg et al. 2013). Since dam removal and channel modification are growing practices in river restoration, it is important to understand ecological processes that take place after habitat such modifications. Filling the void in river restoration research and connecting restoration techniques with specific ecological goals would allow for better informed decisions on the part of watershed managers and other stakeholders.

River restoration projects across the world use a wide variety of techniques depending on the goals of the restoration. Kail et al. (2015) conducted a metadata analysis on peer reviewed studies in order to understand the efficacy of different restoration techniques on river systems. In this study, Kail et al. (2015) suggested that restoration efforts do, in general, have a positive impact on diversity and biomass of fish, macrobenthic, and macrophytic communities, however responses in certain circumstances were incredibly varied and included examples of a net negative effect caused by restoration efforts. These variable responses in river systems may be the reason a scientific consensus on the best forms of restoration efforts has yet to be reached. Kail et al. (2015) went on to show that restoration had varying levels of effectiveness based on the subjects studied with macrophytes seeing the highest improvement and macroinvertebrates seeing the lowest. Instream restoration methods proved to have the greatest positive impacts on macroinvertebrates and fish species as opposed to alteration of the shape of the river itself. They also found that restorations caused a greater impact on the biomass of invertebrate and fish species rather than the diversity and richness, suggesting that new habitats were being colonized by existing species in the immediate area as opposed to allowing for new species to gain a toehold.

The second half of Kail et al. (2015) looked to identify restoration characteristics that proved most influential in causing a positive response by biota. The authors discovered that the percent agricultural land use in the catchment, river width and age of the restoration were the three major characteristics that influenced restoration communities (Kail et al. 2015).

Kail and Wolter (2010) showed that human impacts on river catchments had a significant effect on communities within the river course, with macroinvertebrates being the most effected ahead of fish and macrophyte communities. They found that pressures at the river catchment scale accounted for between 38 and 100% of the variation found in macroinvertebrate while reach-scale pressures (i.e. channel morphology, presence of boulders, etc.) only accounted for up to 8% of the variation in macroinvertebrate communities.

In contradiction to Kail and Wolter (2010); Miller et al. (2010) did not find significant differences in post-restoration biota responses among land use criterion in a meta-analysis study. Interestingly, they did find that restorations that took place in forested regions did have significantly less variable responses compared to other land-use areas. These assessments support that river response trajectories are incredibly complex with likely many confounding factors leading to a lack of scientific consensus within the literature.

Although there have been several restoration projects attempted, there is not a true scientific consensus on if these restoration techniques are accomplishing the overarching goal of transitioning impaired rivers back to a more natural state. Muhar et al. (2016) attributed these conflicting findings to a lack of standardized methodology for restoration studies, while other suggestions included major differences in the scale of restoration and other confounding conditions across the entire catchment in question. Muhar et al. (2007) showed that the success of rehabilitation attempts in ecological restoration was effected by the scale of the restoration effort. They go on to explain that this relationship is likely due to increased areas of important patch types within the river which were a direct result of restored.

Macroinvertebrates act as important intermediate trophic consumers by connecting low level food sources such as detritus, macrophytes, algae and other microorganisms with higher level members of the food web such as fish, birds, and other vertebrates (Hay et al. 2008). This keystone link in lotic systems is vital for the passing of nutrients to higher trophic levels in the food web (Malmqvist 2002). Their importance in the lotic food web suggests that with increased health of the macroinvertebrate trophic group, the overall health of the system would increase.

Biological monitoring was pioneered in response to the Federal Water Pollution Control Act of 1972 (FWPCA) to monitor biological integrity of important water resources throughout the nation. These methods were pioneered by Karr (1981) to efficiently and affordably investigate the magnitude of degradation on rivers and streams through the quantification of community metrics. This process uses community metrics such as species diversity and evenness to compare potentially impacted streams with “natural” or minimally impacted areas. This method has led to the development of several Indices of Biologic Integrity based on community composition of different biota (algae, fishes, and macroinvertebrates) (Melo et al 2015).

Davies and Jackson (2006) have since proposed the Biological Condition Gradient. This gradient uses biological monitoring to classify rivers into a hierarchy of six tiers of human impacts. These tiers range from 1, meaning a natural or native condition, to 6, meaning severe changes in the structure of the biotic community and loss of ecosystem function. This allows natural resource managers to prioritize restorations based on which rivers are most and need of restoration efforts, as well as which rivers are important for increased conservation efforts based on their high quality.

The use of these organisms as indicators of ecosystem relies on the assumption that macrobenthic community composition changes along a continuum of habitat and water quality. In other words, as community metrics such as species richness and diversity diminish, it can be assumed that the habitat and water quality of that ecosystem is, in turn, equally diminished (Kenny et al. 2009). In restoration ecology, these organisms are often used as important indicators of environmental health (Karr et al. 1986) and have been one of the major metrics in in measuring the success of restoration efforts. The Connecticut Department of Energy and Environmental Protection (DEEP) have created a rapid bioassessment protocol using this theory and have identified common taxa which are used to monitor stream health. Sensitive species, those which cannot tolerate human impacts (i.e. ephemoptera, plecoptera, and trichoptera species), are indicative of high quality habitat and water quality. Pollutant tolerant species indicative of poor stream health include chironimidae, simuliidae, and molluscs (CT DEEP 2012).

This thesis focuses on a restoration project conducted at Pond Lily Dam, located in New Haven Connecticut, (Figure 1). The impoundment of the West River behind Pond Lily Dam extended upstream into the town of Woodbridge, Connecticut and covered an estimated four acres of open water and wetlands including a significant percentage of emergent marshes. The Towns of New Haven and Woodbridge, along with the collaboration with the New Haven Land Trust, American Rivers, and the Connecticut Department of Energy & Environmental Protection (CTDEEP) identified the dam for removal and established six major goals for the project:

1. Restore the river habitat to a more natural environment

2. Enable passage of target fish species, including alewife, blueback herring, and American eel
3. Provide flood relief for residents of Woodbridge Flats with additional flood storage created with the removal of Pond Lily dam.
4. Mitigate liability associated with failure of Pond Lily dam via breaching or removal.
5. Maintain or enhance habitat in the project area.
6. Promote recreational use of the Pond Lily Nature Preserve

Milone & MacBroom, Inc. were contracted by the Town of Woodbridge for alternatives analysis in accordance with the National Environmental Policy Act. In these analysis, the suggested alternative for remediation was outlined in 6 main points:

1. Removal of approximately 100ft of the stone spillway to allow for structural stability of river banks as well as enable fish passage.
2. Partially excavate sediment within the impoundment to create a new channel is a fashion that will minimize sediment excavation and return the channel to what is believed to be the historic alignment.
3. Incorporation of natural channel habitat features such as the placement of woody debris, riffle, vegetated bars, and small boulder clusters.
4. Grading of impoundment sediments adjacent to the new channel to create constructed wetland and riparian upland habitats.
5. Protection of the existing berms along Whalley Avenue and across from the spillway near the established parking lot through placement of fill. Maintain the western portion of the spillway as a wall to be graded to provide additional stability and protection.

6. Design the Pond Lily Dam site in a manner compatible with the New Haven Land Trust's vision of an ecological nature preserve.

Detailed plans and sketches are available on file with the State of Connecticut (Milone & MacBroom 2011).

Because of the proliferation of dam removals and river restorations, it is important to understand how benthic communities will respond after restoration efforts are completed. Biomonitoring efforts will allow natural resource managers to track the environments progress from impaired to a more natural state. By establishing a baseline for the first year after dam removal, this project provides a jumping off point as the Pond Lily Pond ecosystem reverts to its natural condition.

The goal of this research project is to assess how macrobenthic communities respond to dam removal and habitat restoration in the Pond Lily Nature Reserve section of the West River in New Haven, CT. The questions guiding the research were:

- Has the Pond Lily Dam removal and creation of a new riffle habitat resulted in the establishment of a macrobenthic community that is like those found in natural riffle habitats?
- How have environmental conditions changed in affected habitats downstream of the dam removal site?
- Are there seasonal differences among the restored and control areas of the river?

Chapter II:
Materials and Methods

In this study, macroinvertebrate communities were surveyed the summer after the removal of Pond Lily Dam in 2016 A total of four sites were surveyed along the West River at the Pond Lily Dam removal site and the Konolds Pond Dam. The Konolds Pond Dam locations were used to try and understand the community structure of microbenthic invertebrates at a dam location. Descriptions of the sample sites (Figure 2) are as follows:

- Site 1- A riffle location immediately downstream of the removed Pond Lily Dam.
- Site 2- A newly established riffle site immediately upstream of the removed Pond Lily Dam.
- Site 3- A riffle location immediately downstream of the Konolds Pond Dam.
- Site 4- Immediately upstream of the Konolds Pond Dam.

All study sites were $150m^2$ in size with dimensions of 15x10m.

Sample Collection

Macrobenthic samples were collected monthly from May 2016 through October 2016. Samples were taken from four sites: as noted above (Figure 1). Directly upstream and downstream of the dam and former dam location, 150 m^2 areas were designated which were divided into a 1m grid coordinate system. Three replicate samples were taken at each site within these areas on each sampling date. Coordinates for each sample based on the 1 m^2 grid were chosen using a random number generator. At each sample location, several environmental variables were measured including temperature, dissolved oxygen, depth, canopy cover, and substrate type. Temperature and dissolved oxygen were measured using a YSI ProDO Professional Series probe, depth was measured using a transect tape, and percent canopy cover was estimated to the nearest 5% within the entire field of view of the observer looking straight up.

Macrobenthic samples were collected using a 0.88 m^2 Hess stream bottom sample with a 1 mm screen. The sampler was inserted into the substrate to a depth that allowed water flow through the front and back windows of the sampler and into the cod end of the collection net. The substrate within the sampler was disturbed by hand for one minute and the sample was collected in the cod end of the sampler. The contents of the net were then transferred into sample jars and stored in 70% ethanol for later identification.

Organisms were separated from substrate materials using a dissecting microscope. Insects were identified to the lowest feasible taxonomic level (usually family or genus), while other organisms were sorted to more general taxonomic groups (amphipods, isopods, bivalves, gastropods, oligochaetes, hirudinea, fishes, and mites). Identification was done using a variety of keys (e.g. McCafferty 1988) and on-line resources.

Data Analysis

Several overall community metrics, as well as differences in community structure were assessed using PRIMER6 & PERMANOVA software (Clarke and Gorley 2006). Community metrics included species richness, the total number of individuals, and the Shannon-Weiner diversity index H' which combines both species richness and evenness of the relative abundances among taxa within a sample. SPSS v23 was used to conduct two-way analysis of variance (ANOVA) to test differences in number of taxa among sites and locations.

Non-metric multidimensional scaling (nMDS) based on community composition using average abundances across replicates, was used to determine differences in community structure among locations and sampling dates. The Bray-Curtis resemblance function was used to quantify pair-wise sample similarities. Environmental data were overlain as a biplot (from a principal component analysis) in order to assess community differences relative to environmental conditions at the four sites. Principal Components Analysis (PCA) was conducted to assess differences in community structure among study site and times and to determine which taxa were contributing most to the observed differences in community structure.

Two-way Analysis of Similarity (ANOSIM) and PERMANOVA (Clarke and Gorley 2006) were conducted to test differences in community structure among dates and groups as well as to test pairwise differences among individual months. The analyses were based on Bray-Curtis similarities and 9999 permutations of the data. Similarity percentage (SIMPER) analysis (Clarke and Gorley 2006) was used to further explore the community differences among sites in order to quantify differences in communities among sites and sampling times, and determine the relative contributions of the taxa to such differences.

Due to sampling difficulties, I was unable collect some environmental data. This missing data were estimated based on other data collected at that site or that day. Since the temperature and dissolved oxygen data that was collected showed negligible differences, missing temperature and dissolved oxygen data was estimated by the mean temperature and dissolved oxygen readings that were measured on the same day at other locations. Missing substrate data was estimated by finding the most frequent substrate type sampled at that location across all of the sample dates. Water velocity data was excluded from analysis due to malfunctions during data collection. For community multivariate analyses, data collected on each day at each site were averaged to find the average community composition and environmental conditions (the mode was used for substrate) for each sampling date. Community data were transformed with a log(x+1) transformation in order to diminish the impact of outliers on the data.

CHAPTER III:
Results

Changes in Diversity and Abundances

A total of 54 taxa were found in the West River between May and October. The highest mean number of taxa was found at site 3 directly downstream of the Konolds pond dam (Figure 3). Taxonomic richness during the sampling period varied greatly from site to site. Site 1 and 2 showed a similar pattern in increasing in richness in the first three weeks, reaching a peak, and then declining (despite week 4 at site 1 dropping suddenly then recovering). Site 2 had a higher peak than site 1 during August with a mean richness of 14.33 and gradually decreased until the end of the sample period. Site 3 showed no pattern in richness across the sampling period however peaked at 15 taxa during July. Site 4 had consistently low numbers of taxa from July onwards with only 6-7 taxa after the May and June when taxonomic richness was substantially higher (Figure 3). A two-way ANOVA indicated that there was no statistically significant difference in taxonomic richness among sampling dates ($p=0.161$), however there was a significant difference among sites and the interaction of sites and dates (Table 1). This difference is caused by the increased taxonomic richness in the summer at the Pond Lily Dam site, likely due to differences in temperature between the two sites.

The Shannon-Wiener diversity was similar during the sampling period at sites 1 and 3, while at site 2 it peaked during August. Site 4 was the only site to show a decease across the sampling period (Figure 4). Site 1 had the highest mean Shannon-Wiener diversity ($H'=1.79$) while site 3 had the lowest ($H'=1.35$). Two-way ANOVA indicates a statistically significantly difference among dates for H' but not among sites, or the interaction of site and date (Table 2).

The four taxa with the largest impact of differences between communities were chironamidae, hydropsychidae, bivalves and gastropods (Figure 5). Chironamidae were consistently one of most abundant taxa at most of the sites. There was significant variation in chironamidae abundance at site 3 across all 6 months sampled. Mollusk populations at this site increased over time while populations of hydropsychidae remained consistent. Hydropsychidae were completely absent from all four sites in the first month of sampling but became a dominant aspect of the community at site 2. Site 4 was strongly impacted by an increasing abundance of bivalves from months 1 to 4 before a sudden drop in bivalve population (Figure 5).

Changes in Community Structure

The Result of a nMDS ordination of community structure indicates a general similarity at sites 3 and 4 across the sampling period, but that these sites were different from sites 1 and 2 which had relatively variable communities over sampling period (Figure 6). This is borne out by the dispersion analysis which shows that sites 1 and 2 had much higher dispersion (variability) indices overall than sites 3 and 4, and also in pairwise tests t (Table 3). The environmental biplot indicates sites 3 and 4 are negatively associated with dissolved oxygen and positively correlated with vegetation cover percentage, ands to a lesser extent higher temperatures and substrate type.

Principal component analysis of the communities shows a similar trend to the nMDS (Figure 7). Sites 1 and 2 appear to be separated from sites 3 and 4 especially across PC1 which accounts for 29.7% of the variation in the data (Table 4). Sites 3 and 4 show some differentiation across PC2 which accounts for 18.1% of the variation in the data (Table 4). The influence of mollusks on each community has the greatest impact on community differences (loadings- PC1- gastropod=0.417, bivalve=0.391). PC2 is most heavily influenced by the presence of hydropsychidae (loadings PC2=0.49) (Table 5). The biplot overlain on the PCA reflects the impact of four major taxa on community composition as measured by the loadings. Sites 1 and 2 have a strong positive correlation with the presence of hydropsychidae while are negatively correlated with the presence of chironamids and gastropods, as opposed to Sites 3 and 4 which are positively correlated with the presence of Gastropods and Bivalve.

Two-way crossed ANOSIM results indicate there is a significant overall difference between sites and dates. Pairwise tests show that each site was statistically different from all of the other locations with the greatest statistical difference between sites 1 and 3, 1 and 4, and 2 and 4. When comparing communities across the sampling period, each date is statistically different from the following date except when comparing August and September (Table 6). Permanova results show there is a significant difference in community structure between dates, sites, and the interaction of site and date (p=0.001, 0.001, 0.001 respectively, Table 7).

The SIMPER analysis reveals what community characteristics caused the differences among dates and locations (Table 8). From month to month, chironamidae and hydropsychidae appear to have the most influence throughout the entire sampling period. Bivalves and gastropods were also large contributors to variations between communities. Within sites, Chironamidae had the highest contribution between samples taken (Site 1=24.81%, Site 2=28.81, Site 3=19.22, Site 4= 32.13).

18 of the 54 taxa accounted for 90.53% of the dissimilarity between sites 3 and 4 (Table 8). These sites shared the lowest dissimilarity of 50.23% due largely to the prevalence of mollusks at site 4, which accounted for more than 21% of that dissimilarity. 19 taxa accounted for 90.97% of the dissimilarity between sites 1 and 4. These sites shared the highest dissimilarity of 75.94% with chironimidae contributed the highest dissimilarity percentage of 11.34%, and oligochaetea contributing 11.34%.

The Pond Lily Dam sites (1 and 2) shared an average dissimilarity of 62.81% (Table 8). chironamidae accounted for 10.52% of the dissimilarity with elmidae, gastropods, and hydropsychidae accounting for the four most dissimilar taxa accounting for 35% of the dissimilarity. Of the 54 taxa found in this study, 22 taxa accounted for 91.38 of the dissimilarity between the Pond Lilly Pond sites.

Communities in May and June shared an average dissimilarity of 53.78%. 15 of the taxa accounted for 90.16% of the dissimilarity among these first two months of sampling. The three greatest contributors to this difference were due to nematocera, chironamidae, and hydropsychidae, accounting for 33.46% of the total dissimilarity. Chironamidae were the biggest contributor to dissimilarity between June and July, July and August, and August and September. All sets of dates required the inclusion of more than 20 taxa in order to account for more than 90% of the difference between those months. September and October were 48.76% dissimilar. Chironamidae contributed 14.13% of that dissimilarity while the next highest contributor were oligochaeta, contributing 7.54% while 20 taxa were needed to account for 90.22% of the dissimilarity (Table 9).

Chapter IV:
Discussion

The goal of this research was to assess changes in the benthic community at the Pond Lily Dam removal site in order to understand how the habitat restoration affected the community composition and dynamics. The Konolds Pond dam area was used as a nominative control site to assess the changes at the Pond Lily Dam site following removal of the dam, and in order to understand what types of communities exist in the area of dams along the West River, and to attempt to see how the communities are changing since the removal of the dam.

Trends in taxonomic richness at the Konolds Pond reference site were highly variable downstream of the impoundment, from month to month (Figure 3). This trend is consistent with the conclusion drawn Bhaumik et al. (2017) which showed that areas down stream of dams are highly variable over time. Upstream of the Konolds Pond Dam, richness decreased between June and July, and remained low throughout the rest of the study period. This loss of richness coincides with warming water temperatures and decreased dissolved oxygen content which is expected during summer months.

The taxonomic richness of the macrobenthic community at Site 1, which was formerly downstream of the Pond Lily Dam impoundment, increased until July, then appeared to stabilize (with the exception of August, an anomaly). Taxonomic richness at Site 2, a newly established riffle area, increased until peaking in August, and then steadily declining throughout the remainder of the study period. The trends at the rehabilitated Pond Lily Dam sites were significantly different from the corresponding upstream/downstream locations at Konolds Pond. These differences suggest a change in community structure from what would normally be found around an impoundment and a shift to a new community structure.

Shannon-Wiener diversity at both downstream locations (Site 1 and 3) appeared to stay somewhat consistent throughout the study (Figure 4). In contrast, the upstream location at Pond Lily (Site 2) followed a similar pattern over time as taxonomic richness, and the Konolds Pond location shows a downward trend in diversity over the study period. These results further supports that community structure at the rehabilitated Pond Lily Dam impoundment area shifted form that which might be found in impoundment areas

Variability in community composition was also revealed by the multivariate analyses. nMDS (Figure 6) shows sites 3 and 4 grouped closely together across most months, indicating fairly similar community structure. Sites 1 and 2 showed much more variability throughout the sampling period which is seen in the dispersion indices (Table 3). This increased variability also suggests that the Lily Pond dam sites are in the process of recolonization/succession in response to the disturbance of the dam removal and have yet to stabilize to a less variable community structure.

Community structure was significantly different among the four sampling sites (Table 6). Since sites 1 and 2 are now connected without any major impediments between them, macrobenthic community structure at the two locations may eventually be similar since the sites are similar riffle habitats. During the study period however, there was a significant difference between the two Sites. Tullos, Finn and Walker (2015) found that macroinvertebrate assemblages at upstream and downstream locations the year after dam removal were not significantly different. Instead, they showed a similar community structure above and below the dam removal site. Other studies suggest that invertebrate assemblages downstream of impoundments remain similar in composition before and after removal of an impoundment (Pollard and Reed 2004). This is not consistent with what was found around the Pond Lily Dam removal highlighting the variability of response trajectories from one restoration effort to the next. PCA (Figure 7) show the impact of chironimidae and gastropods on the separation between the two downstream sites. These two taxa, which are indicative of more impaired water quality, are found to be more abundant downstream of the Konolds Pond impoundment and less abundant downstream of the removed Pond Lily Dam.

The Pond Lily site has undergone significant change since the removal of the dam and establishment of a riffle system. If the Konolds Pond is representative of macrobenthic communities around dams on the West River, we can begin to infer what parts of the community have changed. The prevalence of gastropods and bivalves at the Konolds Pond site plays a major role in the separation between it and the Pond Lily Dam site. The Konolds Pond sites are heavily influenced by the presence of bivalves, gastropods, annelids, chironamidae, and simuliidae. These groups have been identified by the Connecticut DEEP as species indicative of poor water quality suggesting the West River in the Konolds Pond area is impaired (EPA 2007). While the same taxa were present at the Pond Lily Dam site, the prevalence of trichopertans, specifically hydopsychidae, suggest that this location is less impaired than the Konolds Pond site. This is a promising result after only one year following completion of restoration efforts.

River restoration aims at altering environmental conditions to be more favorable for species of interest. This method has been referred to as the “Field of Dreams” hypothesis with the idea that creating abiotic and environmental factors in which a species is found will encourage colonization of those species (Miller et al. 2010). The Pond Lily Dam restoration aimed at creating high quality habitat which would thereby be colonized by species indicative of less impaired streams. The monitoring of these projects, as well as surface water quality, across the nation has been done through a rapid bioassessment protocol which uses macrobenthic communities as a measurement of stream health (Barbour et al. 1999). In these protocols, taxa such as gastropods, chironamidae, and siimuliidae are indicative of lower quality habitat while orders ephemeroptera and neuroptera are indicative of high quality habitat. Other taxa such as Odonata and Hydropsychidae are indicative of moderate habitat quality (EPA 2007).

Primary components analysis showed a significant separation across PC1 which accounts for 29% of the variation (Figure 7). This primary component is significantly impacted by the presence of bivalves and gastropods as suggested in Table 5. The high number of bivalves and gastropods at both Konolds Pond locations and low numbers at the Pond Lily Dam site suggest that the Konolds Pond Dam is less favorable habitat for these taxa. Since the two locations are so close in proximity, if we assume the Pond Lily dam ecosystem was similar to that of the Konolds Pond ecosystem, we can assume the Pond Lily Dam ecosystem was also heavily impacted. Across PC2, hydropsychidae played a large role. The presence of hydropsychidae at the Pond Lily Dam suggests that conditions at the once heavily impacted site are now ameliorated and that there is a positive ecosystem response one year after restoration efforts were completed.

Bellucci et al. (2011) used the abundance of ephemeroptera, plecoptera, and trichoperta (EPT) taxa as a metric of habitat quality in Connecticut streams because these orders are known to be a dominate component of community richness in least disturbed conditions whereas increased abundance of diptera and mollusk taxa suggest increased disturbance in the area. In this study, the restored Pond Lily sites were dominated by hydropsychidae whereas the Konolds Pond sites were dominated by gastropods, bivalves, and simuliidae. This shows that restoration has created a less stressed environment at the Pond Lily sites.

While there appears to be a positive response to restoration in this situation, a complete restoration of the habitat is unlikely. Colonization of recently disturbed stream habitats are highly dependent on source populations that have the potential of moving into the new habitat. Westveer et al. (2017) discussed that source population characteristics, such as distance and dispersal mechanism, play the major role in how and when recolonization takes place. They point out that the recolonization of stream habitats is based largely on the upstream communities as the source population since the flow of the stream is the driving dispersal mechanism of the habitat. In order for a restored stream to reach a potential "natural" state, recolonization would have to be accomplished by a nearby "natural" source population, which, in the case of the West River, is not available due to multiple impoundments upstream of the restored section of the West River which impede dispersion. In order to accomplish that feat, larger-scale restoration would have to be considered focusing on a catchment-scale approach to stream restoration.

Future Research

In this study, I established a baseline for the benthic macroinvertebrate communities that exist at the Pond Lily Dam restoration site one year following the dam removal and river restoration project. To understand the efficacy of this river restoration effort, long term monitoring is necessary. While there is some evidence to suggest the Pond Lily Dam section of the West River is improving, community differences between the two Pond Lily Dam segments suggest a climax community has not yet been established. Sites 1 and 2 are only about 20 yards apart and are both riffle sections and my expectation is they would have very similar communities when a climax community is reached. Such a climax community would be established as the newly restored areas reach the natural dynamic equilibrium found in the hydrology of streams. Invertebrate communities in undisturbed streams in Connecticut are characterized by greater abundance of EPT taxa (Bellucci, Becker, and Beauchene 2013).

Community patterns in the restored section of the West River showed increasing amounts of these species suggesting a habitat less influenced by human impacts.

Reach-scale studies are valuable for understanding the effects of individual rehabilitation efforts such as dam removal. Additional studies on how different restoration techniques can shed light on specific responses which can inform decisions for future restoration efforts. However, it is not a complete picture of the condition of a river or other human pressures that can be altered to better the restoration effort. Since rivers are closely tied to the landscape that they flow through, a catchment scale understanding of land use trends would be beneficial for managers to better restore ecosystems.

Conclusions

Worldwide, dam removal is being used as an important tool for restoring degraded rivers (Schiff, Benoit and Macbroom 2011). While there is an apparent belief that these restorations create valuable habitat for native species, there has been little documented proof in the literature to support that idea (Muhar et al. 2016). This study aimed at understanding how dam removal and river restoration projects effect macrobenthic invertebrate communities, an important group for surface water quality and habitat monitoring in the United States. The removal of the Pond Lily dam and the restoration of its impoundment to a natural stream area appear to have had beneficial effects on habitat quality within the first year after restoration was completed. While some positive effects have been seen, these communities were incredibly dynamic throughout the sampling season and appear to have not reached any type of climax community. Continuous monitoring of these restoration efforts are important in order to ensure the long-term response trajectories of these ecosystems match long-term goals of decision makers.

Figures and Tables:

Figure 1- Site Location. (Top Left- Pond Lily Lower, Top Right- Pond Lily Upper, Botom Left- Konolds Pond Lower, Bottom Right- Konolds Pond Upper.)

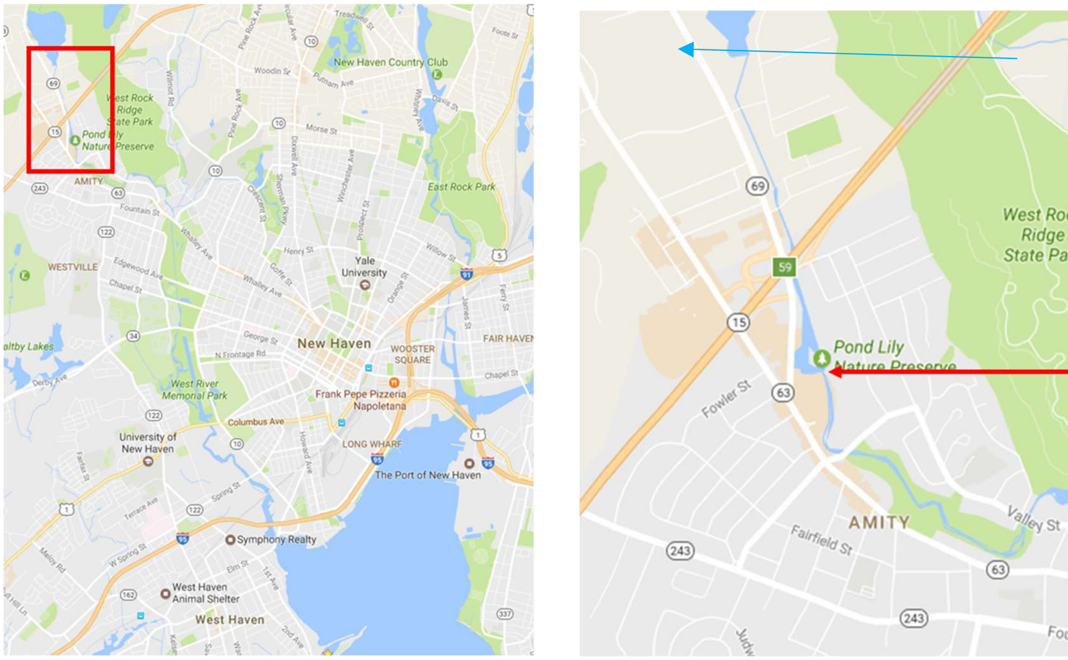

Figure 1. Left: Map of the New Haven area. Pond Lily Nature Preserve and Konolds pond dam location highlighted in red. Right: Location of Pond Lily Dam (Red) (Sites 1 and 2) and Konolds Pond Dams (Blue) (Sites 3 and 4).

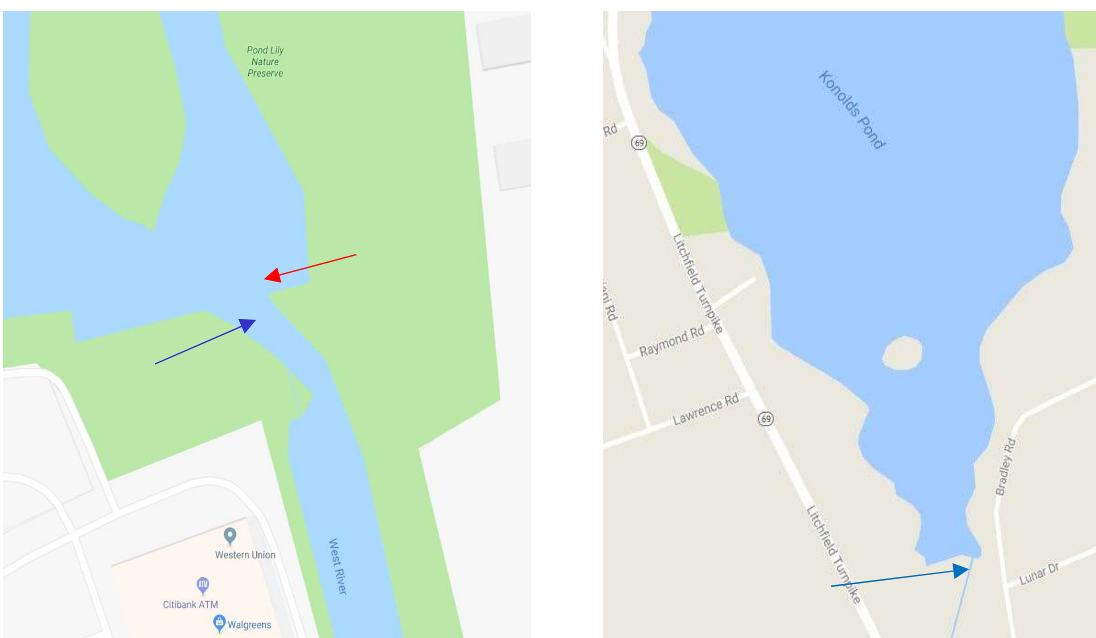

Figure 2. Left: Pond Lily Dam location. Blue Arrow- Site 1, Red Arrow, Site 2. Left: Map of Konolds Pond. Blue Arrow-Site 3, Red Arrow- Site 4

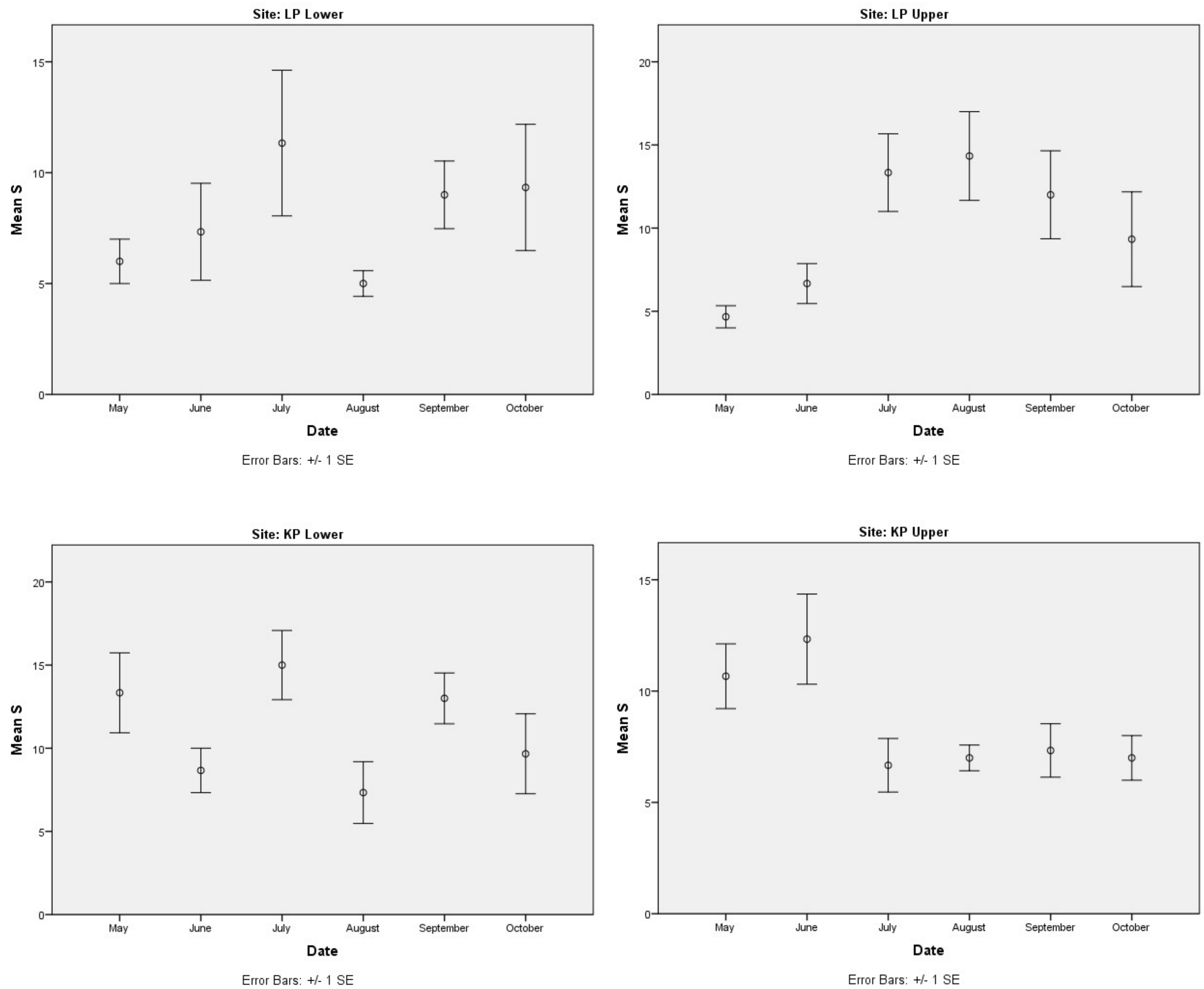

Figure 3. Mean taxonomic richness across the six sampling months at each location +/- 1 SE (clockwise from top left: Site 1, Site 2, Site 4, Site 3).

Table 1. Results of two-way ANOVA testing differences taxonomic richness among sites and dates

Tests of Between-Subjects Effects

Dependent Variable: S

Source	Type III Sum of Squares	df	Mean Square	F	Sig.
Corrected Model	640.986[a]	23	27.869	2.456	.004
Intercept	6403.347	1	6403.347	564.310	.000
Site	113.708	3	37.903	3.340	.027
Date	94.569	5	18.914	1.667	.161
Site * Date	432.708	15	28.847	2.542	.007
Error	544.667	48	11.347		
Total	7589.000	72			
Corrected Total	1185.653	71			

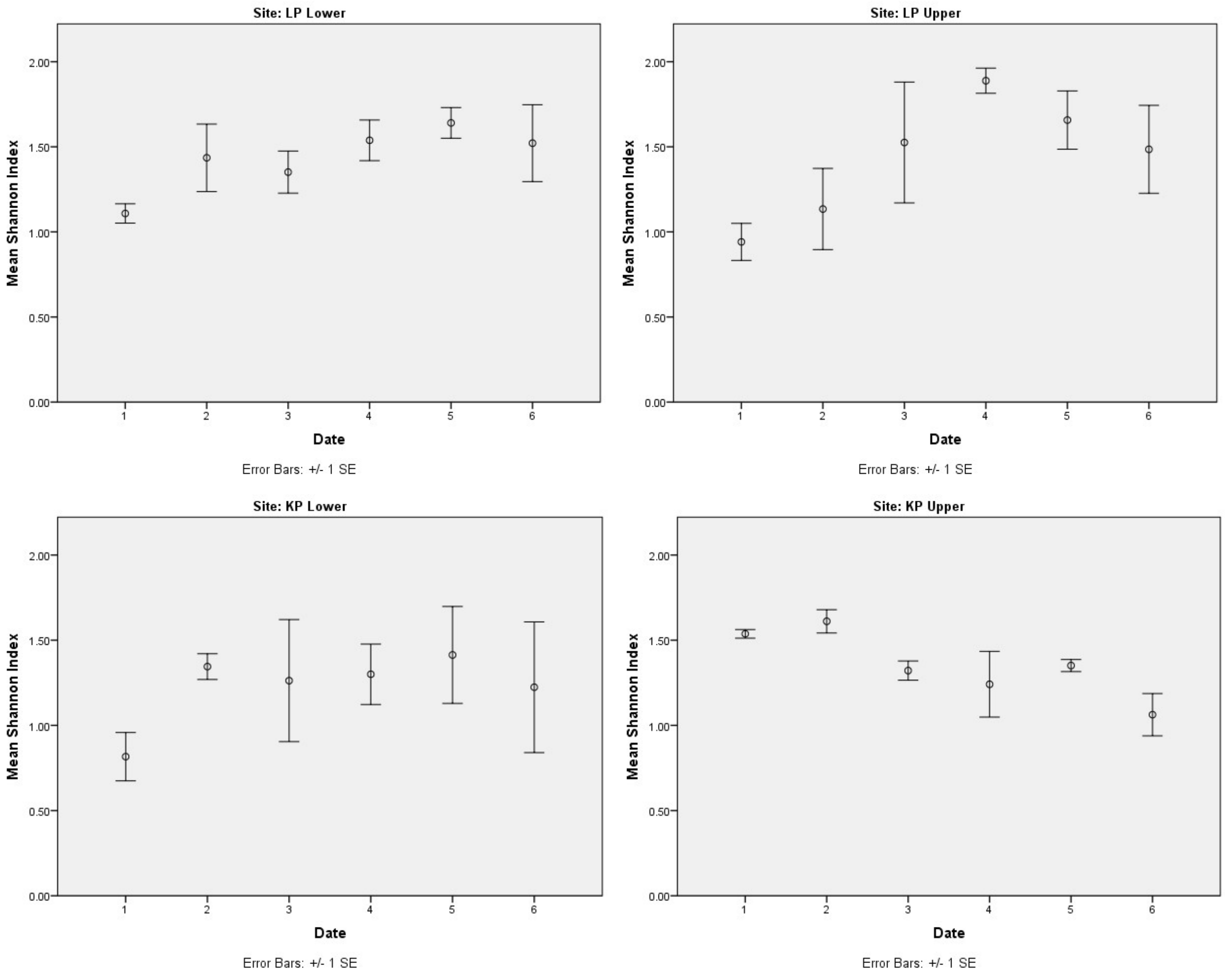

Figure 4. Mean Shannon-Wiener H' index across the six sampling months at each location +/- 1SE (clockwise from top left: Site 1, Site 2, Site 4, Site 3).

Table 2. Results of two-way ANOVA testing differences in Shannon-Weiner H' values between sites and dates

Tests of Between-Subjects Effects

Dependent Variable: Shannon

Source	Type III Sum of Squares	df	Mean Square	F	Sig.
Corrected Model	4.112[a]	23	.179	1.583	.090
Intercept	133.781	1	133.781	1184.399	.000
Date	1.327	5	.265	2.349	.055
Site	.524	3	.175	1.547	.214
Date * Site	2.261	15	.151	1.335	.220
Error	5.422	48	.113		
Total	143.315	72			
Corrected Total	9.534	71			

a. R Squared = .541 (Adjusted R Squared = .320)

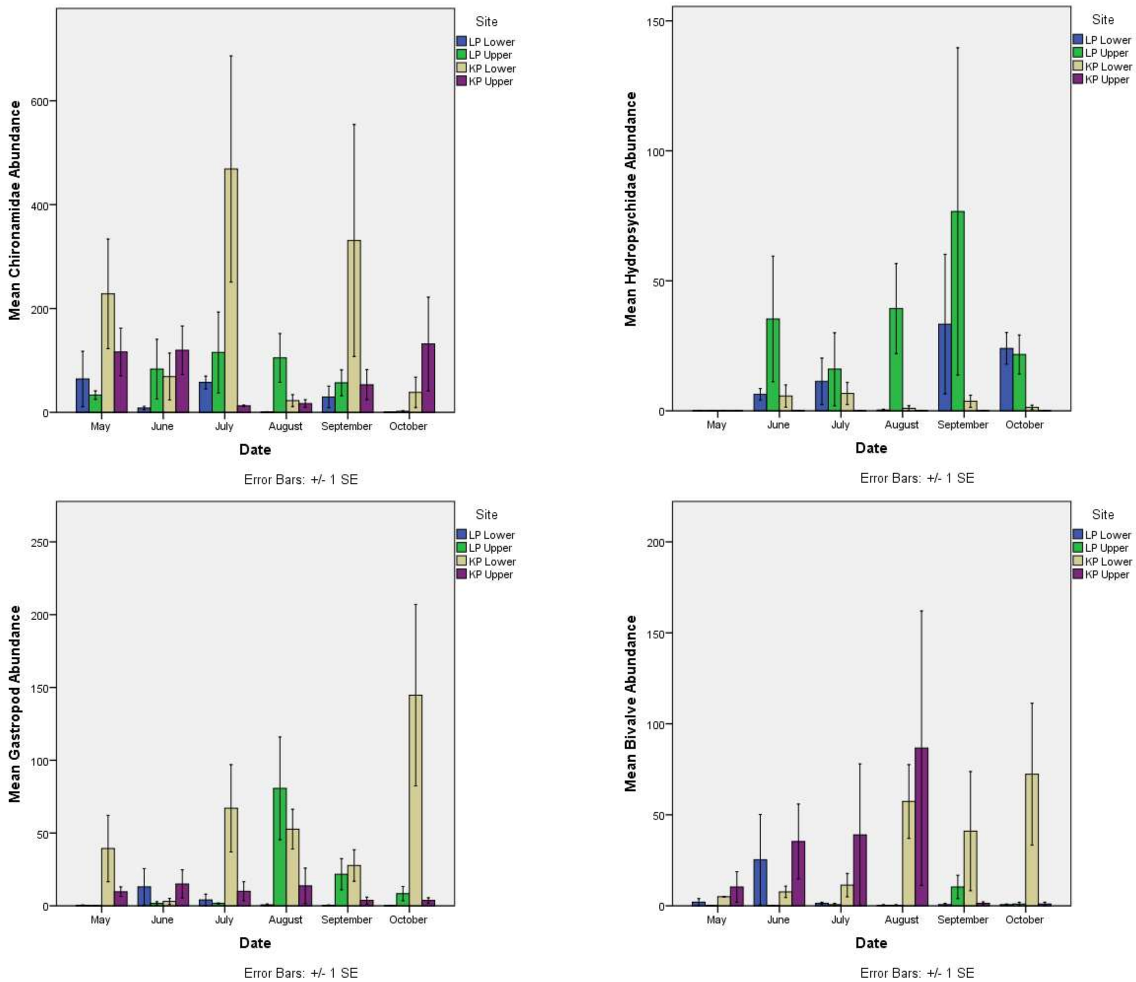

Figure 5. Mean abundance of the four taxa with the largest effects on community difference (per $0.88m^2$ +/- 1SE). (Clockwise from top left: Chironamidae, Hydropsychidae, Bivalves, Gastropods)

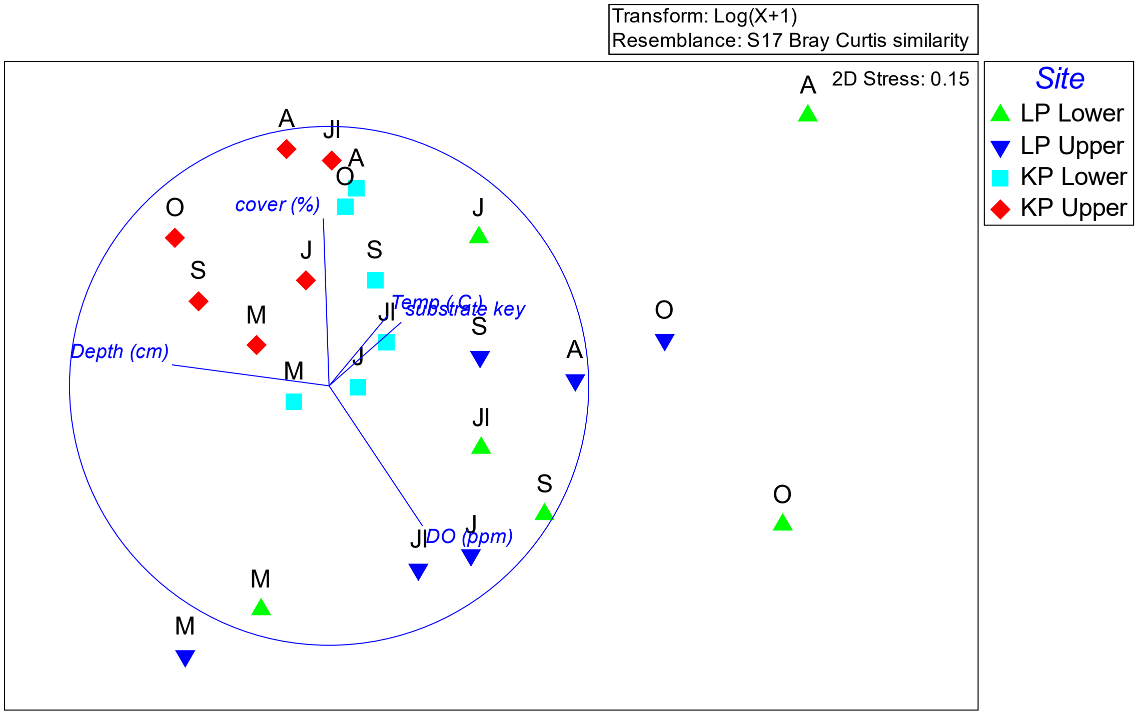

Figure 6. Results of nMDS showing groupings of similar communities at the family level at each site across each date sampled with environmental factors overlain. Two-dimensional stress = 0.15, suggesting a reasonable goodness of fit for the data.

Table 3. Multivariate Dispersion indices of the sample locations

Global Analysis

Site	Dispersion
3	0.673
4	0.693
2	1.231
1	1.403

Pairwise Comparisons (IMD- Index of Multivariate Dispersion)

Site	IMD
1, 2	0.173
1, 3	0.742
1, 4	0.724
2, 3	0.547
2, 4	0.564
3, 4	-0.04

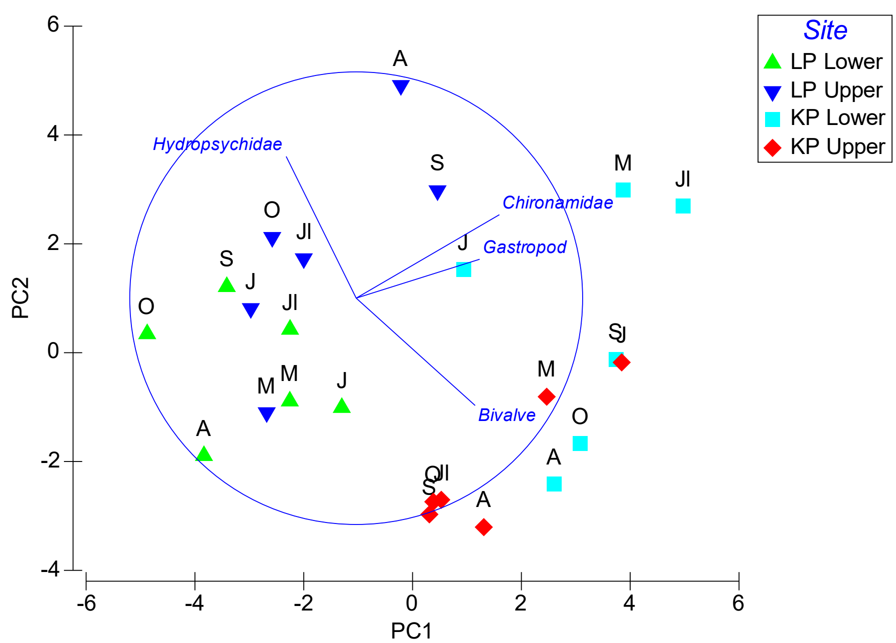

Figure 7. Result of PCA of communities at the family level at each site across each date sampled with the species biplot overlain. Numbers above the symbols indicate sampling month.

Table 4. Eigenvalues from primary component analysis.

PC	Eigenvalues	%Variation	Cumulative %Variation
1	7.94	29.7	29.7
2	4.84	18.1	47.7
3	4.54	17.0	64.7
4	2.59	9.7	74.4
5	1.42	5.3	79.7

Table 5. Loadings of the 5 most influential families on the 2 primary components.

Variable	PC1	Variable	PC2
Gastropod	0.417	Hydropsychidae	0.49
Bivalve	0.391	Amphipod	0.424
Oligicheate	0.378	Simuliidae	0.296
Chironamidae	0.351	Hydroptilidae	0.273
Isopod	0.334	Chironamidae	0.204

Table 6. Two-way Crossed ANOSIM showing differences between sites (Global R=0.572, p=0.01) and dates (Global R=0.451, p=0.01)

Pairwise Tests between sites

Sites	R Statistic	Significance Level %	Possible Permutations	Actual Permutations	Number >= Observed
1, 2	0.355	0.2	1000000	9999	18
1, 3	0.599	0.01	1000000	9999	0
1, 4	0.75	0.01	1000000	9999	0
2, 3	0.605	0.02	1000000	9999	1
2, 4	0.753	0.01	1000000	9999	0
3, 4	0.506	0.03	1000000	9999	2

Pairwise Tests between dates

Date	R Statistic	Significance Level %	Possible Permutations	Actual Permutations	Number >= Observed
1, 2	0.287	0.7	10000	9999	71
1, 3	0.639	0.06	10000	9999	5
1, 4	0.759	0.03	10000	9999	2
1, 5	0.583	0.08	10000	9999	7
1, 6	1	0.01	10000	9999	0
2, 3	0.306	0.6	10000	9999	57
2, 4	0.343	1.1	10000	9999	105
2, 5	0.287	1.2	10000	9999	122
2, 6	0.574	0.07	10000	9999	6
3, 4	0.463	0.2	10000	9999	17
3, 5	0.259	3.7	10000	9999	372
3, 6	0.722	0.01	10000	9999	0
4, 5	0.185	8.1	10000	9999	805
4, 6	0.185	11.9	10000	9999	1184
5, 6	0.213	3.7	10000	9999	366

Table 7. Results of PERMANOVA

Source	df	SS	MS	Pseudo-F	P(perm)	Unique perms
Site	3	40635	13545	10.904	0.0001	9900
Date	5	23324	4664.7	3.7551	0.0001	9866
Site x date	15	36941	2462.7	1.9825	0.0001	9810
Res	48	59628	1242.3			
Total	71	1.6053E5				

Table 8. Results of similarity percentage analysis (SIMPER). The table shows the percent contribution of each taxa to the total similarity within a study area and the dissimilarity between study areas. Av.Abund= average abundance per sample at that location; Av.Sim= average similarity among replicates at that location; Sim/SD= Similarity standard deviation; Contrib%= percent contribution wo within site similarity; Cum%= Cumulative similarity; Av.Diss= Average dissimilarity between sites; Diss/SD= Dissimilarity standard deviation.

Site 1
Average similarity: 42.12

Species	Av.Abund	Av.Sim	Sim/SD	Contrib%	Cum.%
Chironamidae	2.11	10.45	1.20	24.81	24.81
Elmidae	1.87	9.96	1.61	23.65	48.46
Hydropsychidae	1.63	8.51	0.94	20.21	68.67
Nematocera(pupa)	0.68	2.77	0.64	6.58	75.25
Platyhelminthes	0.49	2.39	0.59	5.67	80.92
Fish	0.42	1.61	0.24	3.81	84.73
Amphipod	0.31	1.29	0.50	3.07	87.80
Simuliidae	0.47	1.28	0.50	3.03	90.83

Site 2
Average similarity: 51.55

Species	Av.Abund	Av.Sim	Sim/SD	Contrib%	Cum.%
Chironamidae	3.25	14.85	1.21	28.81	28.81
Hydropsychidae	2.39	8.14	1.17	15.79	44.59
Nematocera(pupa)	1.21	6.14	0.63	11.90	56.50
Gastropod	1.72	4.80	1.05	9.31	65.80
Hydroptilidae	1.30	4.01	0.71	7.77	73.57
Amphipod	1.55	3.47	0.69	6.72	80.30
Simuliidae	0.52	2.71	0.47	5.26	85.56
Philopotamidae	0.54	1.00	0.42	1.93	87.49
Bivalve	0.52	0.90	0.42	1.75	89.24
Water mite	0.45	0.90	0.34	1.74	90.99

Site 3
Average similarity: 63.51

Species	Av.Abund	Av.Sim	Sim/SD	Contrib%	Cum.%
Chironamidae	4.21	12.21	2.79	19.22	19.22
Gastropod	3.38	11.85	1.65	18.66	37.88
Bivalve	2.78	9.99	1.66	15.72	53.60
Oligochaeta	2.43	6.90	1.46	10.87	64.47
Simuliidae	2.00	5.42	0.71	8.54	73.01
Hirudinidae	1.77	4.67	0.91	7.36	80.37
Amphipod	1.89	4.06	1.30	6.40	86.77

Isopod	1.75	2.69	0.69	4.23	91.00

Site 4
Average similarity: 58.35

Species	Av.Abund	Av.Sim	Sim/SD	Contrib%	Cum.%
Chironamidae	3.72	18.75	4.03	32.13	32.13
Oligochaeta	2.35	10.56	1.22	18.10	50.23
Gastropod	1.78	5.64	1.18	9.66	59.89
Ceratopogonidae	1.65	4.88	0.68	8.36	68.25
Hirudinidae	1.22	4.73	0.76	8.11	76.36
Isopod	1.16	3.70	0.83	6.35	82.71
Bivalve	1.74	3.63	0.65	6.22	88.94
Amphipod	1.09	2.67	0.68	4.58	93.52

Site 1 & 2
Average dissimilarity = 62.81

	Group 1	Group 2				
Species	Av.Abund	Av.Abund	Av.Diss	Diss/SD	Contrib%	Cum.%
Chironamidae	2.11	3.25	6.61	1.34	10.53	10.53
Elmidae	1.87	0.46	5.73	1.36	9.12	19.65
Gastropod	0.52	1.72	5.37	1.29	8.55	28.20
Hydropsychidae	1.63	2.39	4.82	1.05	7.68	35.88
Amphipod	0.31	1.55	3.77	0.98	6.01	41.89
Hydroptilidae	0.22	1.30	3.65	0.88	5.81	47.69
Bivalve	0.69	0.52	2.95	0.76	4.70	52.39
Nematocera(pupa)	0.68	1.21	2.65	1.01	4.22	56.61
Oligochaeta	0.33	0.57	2.45	0.80	3.90	60.51
Fish	0.42	0.04	2.04	0.37	3.25	63.76
Philopotamidae	0.25	0.54	1.97	0.63	3.13	66.89
Platyhelminthes	0.49	0.20	1.82	0.70	2.89	69.78
Simuliidae	0.47	0.52	1.66	0.77	2.64	72.42
Isopod	0.35	0.33	1.60	0.81	2.54	74.96
Odontoceridae	0.06	0.58	1.51	0.56	2.41	77.38
Hirudinidae	0.42	0.16	1.51	0.61	2.40	79.78
Water mite	0.13	0.45	1.36	0.56	2.17	81.95
Tricorythidae	0.27	0.39	1.33	0.88	2.12	84.07
Tipulidae	0.18	0.37	1.21	0.57	1.92	85.99
Empididae	0.04	0.38	1.17	0.56	1.86	87.85
Brachycera(pupa)	0.17	0.30	1.14	0.63	1.81	89.66
Brachycentridae	0.13	0.15	1.08	0.52	1.72	91.38

Site 1 & 3
Average dissimilarity = 69.84

Species	Group 1 Av.Abund	Group 3 Av.Abund	Av.Diss	Diss/SD	Contrib%	Cum.%
Gastropod	0.52	3.38	9.58	1.57	13.72	13.72
Bivalve	0.69	2.78	7.74	1.34	11.08	24.80
Chironamidae	2.11	4.21	6.47	1.59	9.27	34.07
Oligochaeta	0.33	2.43	6.35	1.11	9.09	43.16
Hirudinidae	0.42	1.77	4.80	1.19	6.87	50.03
Simuliidae	0.47	2.00	4.35	0.77	6.23	56.26
Elmidae	1.87	0.40	4.19	1.35	6.00	62.27
Isopod	0.35	1.75	4.18	1.19	5.99	68.26
Amphipod	0.31	1.89	3.92	1.27	5.61	73.87
Hydropsychidae	1.63	0.90	3.49	1.00	4.99	78.86
Nematocera(pupa)	0.68	1.12	1.87	1.01	2.68	81.54
Platyhelminthes	0.49	0.25	1.16	0.71	1.66	83.20
Fish	0.42	0.00	1.07	0.36	1.53	84.73
Hydroptilidae	0.22	0.19	0.93	0.58	1.33	86.06
Brachycera(pupa)	0.17	0.29	0.82	0.59	1.17	87.23
Water mite	0.13	0.29	0.81	0.54	1.16	88.38
Odontoceridae	0.06	0.26	0.80	0.50	1.14	89.53
Tricorythidae	0.27	0.04	0.76	0.61	1.09	90.61

Site 2 & 3
Average dissimilarity = 61.57

Species	Group 2 Av.Abund	Group 3 Av.Abund	Av.Diss	Diss/SD	Contrib%	Cum.%
Bivalve	0.52	2.78	6.23	1.53	10.11	10.11
Gastropod	1.72	3.38	5.29	1.11	8.59	18.71
Chironamidae	3.25	4.21	4.89	1.19	7.94	26.64
Hydropsychidae	2.39	0.90	4.57	1.10	7.42	34.06
Amphipod	1.55	1.89	4.41	1.48	7.17	41.23
Oligochaeta	0.57	2.43	4.38	1.36	7.11	48.34
Simuliidae	0.52	2.00	4.33	0.72	7.03	55.36
Isopod	0.33	1.75	4.14	1.11	6.73	62.09
Hirudinidae	0.16	1.77	3.67	1.35	5.95	68.04
Hydroptilidae	1.30	0.19	2.74	0.85	4.45	72.49
Nematocera(pupa)	1.21	1.12	1.76	0.96	2.86	75.35
Elmidae	0.46	0.40	1.44	1.13	2.35	77.69
Odontoceridae	0.58	0.26	1.39	0.69	2.25	79.95
Water mite	0.45	0.29	1.31	0.67	2.12	82.07
Philopotamidae	0.54	0.00	1.08	0.50	1.75	83.81
Tipulidae	0.37	0.16	0.96	0.68	1.55	85.37
Tricorythidae	0.39	0.04	0.93	0.81	1.52	86.89

Brachycera(pupa)	0.30	0.29	0.78	0.72	1.27	88.16
Platyhelminthes	0.20	0.25	0.78	0.69	1.27	89.42
Ceratopogonidae	0.08	0.30	0.73	0.66	1.19	90.61

Site 1 & 4
Average dissimilarity = 75.94

	Group 1	Group 4				
Species	Av.Abund	Av.Abund	Av.Diss	Diss/SD	Contrib%	Cum.%
Chironamidae	2.11	3.72	8.61	1.51	11.34	11.34
Oligochaeta	0.33	2.35	8.04	1.60	10.59	21.93
Elmidae	1.87	0.00	6.24	1.96	8.22	30.15
Bivalve	0.69	1.74	5.91	0.94	7.78	37.93
Ceratopogonidae	0.04	1.65	5.72	1.02	7.54	45.47
Hydropsychidae	1.63	0.00	5.50	1.13	7.24	52.71
Gastropod	0.52	1.78	5.36	1.47	7.05	59.76
Hirudinidae	0.42	1.22	4.08	1.24	5.38	65.14
Amphipod	0.31	1.09	3.81	1.15	5.02	70.16
Isopod	0.35	1.16	3.61	1.06	4.76	74.91
Platyhelminthes	0.49	0.15	2.28	0.81	3.01	77.92
Nematocera(pupa)	0.68	0.43	1.79	0.82	2.35	80.28
Corixidae	0.00	0.35	1.64	0.45	2.16	82.43
Simuliidae	0.47	0.51	1.60	0.66	2.10	84.54
Fish	0.42	0.00	1.27	0.37	1.67	86.20
Hydroptilidae	0.22	0.00	0.97	0.49	1.28	87.48
Philopotamidae	0.25	0.10	0.93	0.52	1.23	88.71
Tricorythidae	0.27	0.00	0.92	0.59	1.22	89.93
Water mite	0.13	0.24	0.79	0.45	1.03	90.97

Site 2 & 4
Average dissimilarity = 71.72

	Group 2	Group 4				
Species	Av.Abund	Av.Abund	Av.Diss	Diss/SD	Contrib%	Cum.%
Hydropsychidae	2.39	0.00	6.47	1.45	9.02	9.02
Amphipod	1.55	1.09	5.90	1.71	8.23	17.25
Chironamidae	3.25	3.72	5.72	1.29	7.97	25.23
Oligochaeta	0.57	2.35	5.70	1.25	7.95	33.17
Bivalve	0.52	1.74	5.16	1.13	7.20	40.38
Gastropod	1.72	1.78	5.00	1.52	6.97	47.35
Ceratopogonidae	0.08	1.65	4.81	1.03	6.71	54.06
Isopod	0.33	1.16	3.69	1.00	5.15	59.21
Hydroptilidae	1.30	0.00	3.39	0.88	4.73	63.94
Hirudinidae	0.16	1.22	3.34	1.09	4.66	68.60
Nematocera(pupa)	1.21	0.43	2.81	1.16	3.92	72.52
Simuliidae	0.52	0.51	1.91	0.69	2.66	75.18

Philopotamidae	0.54	0.10	1.46	0.62	2.04	77.22
Odontoceridae	0.58	0.00	1.41	0.58	1.96	79.18
Water mite	0.45	0.24	1.27	0.64	1.77	80.95
Elmidae	0.46	0.00	1.23	0.61	1.72	82.67
Tipulidae	0.37	0.08	1.13	0.64	1.58	84.25
Brachycera(pupa)	0.30	0.15	1.08	0.59	1.51	85.76
Corixidae	0.04	0.35	1.08	0.63	1.51	87.26
Tricorythidae	0.39	0.00	1.04	0.74	1.46	88.72
Empididae	0.38	0.00	1.02	0.47	1.42	90.15

Site 3 & 4
Average dissimilarity = 50.23

	Group 3	Group 4				
Species	Av.Abund	Av.Abund	Av.Diss	Diss/SD	Contrib%	Cum.%
Gastropod	3.38	1.78	5.48	1.17	10.92	10.92
Bivalve	2.78	1.74	5.27	1.46	10.49	21.41
Chironamidae	4.21	3.72	4.65	1.31	9.26	30.68
Amphipod	1.89	1.09	3.96	1.58	7.88	38.56
Oligochaeta	2.43	2.35	3.84	1.20	7.65	46.21
Ceratopogonidae	0.30	1.65	3.75	1.18	7.46	53.67
Simuliidae	2.00	0.51	3.30	0.79	6.57	60.24
Isopod	1.75	1.16	3.09	0.99	6.15	66.39
Hirudinidae	1.77	1.22	2.87	1.25	5.71	72.10
Nematocera(pupa)	1.12	0.43	2.15	1.05	4.28	76.38
Hydropsychidae	0.90	0.00	2.00	0.98	3.98	80.36
Water mite	0.29	0.24	0.97	0.60	1.93	82.29

Table 8 Continued

Corixidae	0.00	0.35	0.96	0.48	1.91	84.20
Elmidae	0.40	0.00	0.79	0.83	1.57	85.77
Platyhelminthes	0.25	0.15	0.70	0.63	1.40	87.17
Brachycera(pupa)	0.29	0.15	0.63	0.52	1.25	88.42
Odontoceridae	0.26	0.00	0.55	0.43	1.09	89.51
Sisyridae	0.04	0.19	0.51	0.48	1.02	90.53

Table 9. Examines Date groups
(across all Site groups)
Date 1
Average similarity: 67.89

Species	Av.Abund	Av.Sim	Sim/SD	Contrib%	Cum.%
Chironamidae	4.09	21.28	2.14	31.34	31.34
Nematocera(pupa)	1.94	11.03	1.09	16.25	47.59
Simuliidae	2.57	10.21	1.32	15.03	62.62
Gastropod	1.45	4.10	0.91	6.04	68.66

Isopod	1.46	3.90	0.79	5.74	74.40
Elmidae	0.78	3.76	0.66	5.55	79.95
Ceratopogonidae	0.94	3.34	0.55	4.92	84.87
Fish	0.63	2.41	0.29	3.55	88.42
Amphipod	0.95	2.20	0.60	3.25	91.66

Date 2
Average similarity: 45.42

Species	Av.Abund	Av.Sim	Sim/SD	Contrib%	Cum.%
Chironamidae	3.35	13.78	2.35	30.33	30.33
Hydropsychidae	1.50	6.73	0.92	14.83	45.16
Bivalve	1.67	4.71	0.89	10.37	55.52
Simuliidae	1.58	4.03	0.81	8.87	64.39
Amphipod	1.27	3.45	0.90	7.60	71.99
Isopod	1.25	2.98	0.76	6.57	78.56
Gastropod	1.40	2.85	0.89	6.27	84.83
Oligochaeta	0.86	1.77	0.49	3.90	88.72
Hirudinidae	1.00	1.66	0.53	3.65	92.37

Date 3
Average similarity: 53.25

Species	Av.Abund	Av.Sim	Sim/SD	Contrib%	Cum.%
Chironamidae	4.06	17.62	2.61	33.10	33.10
Gastropod	1.95	6.25	1.28	11.74	44.83
Hirudinidae	1.56	6.16	0.85	11.57	56.41
Elmidae	0.92	3.61	0.63	6.77	63.18
Oligochaeta	1.12	3.41	1.03	6.40	69.57
Isopod	1.23	2.44	0.65	4.58	74.16
Nematocera(pupa)	0.95	2.36	0.84	4.43	78.58
Hydropsychidae	1.31	2.07	0.65	3.88	82.46
Amphipod	1.16	1.71	0.55	3.22	85.68
Simuliidae	0.95	1.60	0.63	3.00	88.68
Bivalve	1.23	1.57	0.66	2.95	91.63

Date 4
Average similarity: 46.25

Species	Av.Abund	Av.Sim	Sim/SD	Contrib%	Cum.%
Chironamidae	2.49	8.59	1.24	18.58	18.58
Gastropod	2.49	8.05	0.96	17.41	36.00
Bivalve	1.86	6.95	0.81	15.02	51.02
Hirudinidae	1.23	5.04	0.83	10.90	61.92
Oligochaeta	1.46	4.49	0.85	9.71	71.64
Hydropsychidae	1.04	2.32	0.53	5.02	76.65

Hydroptilidae	0.88	1.72	0.53	3.72	80.37
Amphipod	1.08	1.69	0.53	3.65	84.02
Philopotamidae	0.66	1.49	0.54	3.23	87.25
Elmidae	0.34	1.45	0.32	3.14	90.39

Date 5
Average similarity: 52.28

Species	Av.Abund	Av.Sim	Sim/SD	Contrib%	Cum.%
Chironamidae	3.86	14.63	4.25	27.98	27.98
Oligochaeta	2.41	7.75	0.93	14.82	42.80
Hydropsychidae	1.83	5.77	0.92	11.04	53.85
Gastropod	1.89	5.41	1.11	10.35	64.20
Nematocera(pupa)	1.29	3.91	1.49	7.48	71.68
Bivalve	1.47	3.04	0.91	5.81	77.48
Amphipod	1.48	2.95	0.76	5.65	83.13
Elmidae	0.72	2.58	0.58	4.93	88.06
Ceratopogonidae	0.93	1.97	0.34	3.77	91.83

Date 6
Average similarity: 58.21

Species	Av.Abund	Av.Sim	Sim/SD	Contrib%	Cum.%
Hydropsychidae	1.71	9.89	0.93	16.99	16.99
Chironamidae	2.10	8.48	0.84	14.57	31.56
Oligochaeta	1.78	8.27	0.79	14.20	45.77
Gastropod	1.94	7.10	0.74	12.20	57.96
Amphipod	1.32	5.24	1.12	9.00	66.96
Bivalve	1.35	4.57	0.60	7.85	74.81
Elmidae	0.68	3.47	0.54	5.96	80.77
Hydroptilidae	0.76	3.45	0.55	5.93	86.71
Platyhelminthes	0.43	2.16	0.53	3.71	90.41

Dates 1 & 2
Average dissimilarity = 53.78

	Group 1	Group 2				
Species	Av.Abund	Av.Abund	Av.Diss	Diss/SD	Contrib%	Cum.%
Nematocera(pupa)	1.94	0.51	6.19	1.03	11.50	11.50
Chironamidae	4.09	3.35	5.97	0.92	11.10	22.59
Hydropsychidae	0.00	1.50	5.84	0.98	10.86	33.46
Simuliidae	2.57	1.58	5.33	1.10	9.92	43.38
Gastropod	1.45	1.40	3.65	1.12	6.78	50.16
Fish	0.63	0.00	3.23	0.45	6.00	56.15
Bivalve	1.02	1.67	2.71	0.70	5.04	61.19
Oligochaeta	0.89	0.86	2.70	1.18	5.02	66.21
Elmidae	0.78	0.67	2.50	0.67	4.65	70.86

Isopod	1.46	1.25	2.36	1.00	4.38	75.24
Hirudinidae	0.33	1.00	2.19	0.86	4.07	79.31
Brachycera(pupa)	0.66	0.00	1.98	0.85	3.69	83.00
Amphipod	0.95	1.27	1.73	0.80	3.21	86.21
Ceratopogonidae	0.94	0.30	1.38	0.60	2.56	88.77
Tipulidae	0.00	0.21	0.75	0.45	1.39	90.16

Dates 2 & 3
Average dissimilarity = 56.54

Species	Group 2 Av.Abund	Group 3 Av.Abund	Av.Diss	Diss/SD	Contrib%	Cum.%
Chironamidae	3.35	4.06	6.56	1.27	11.60	11.60
Bivalve	1.67	1.23	3.94	0.90	6.97	18.57
Gastropod	1.40	1.95	3.91	1.31	6.92	25.48
Hydropsychidae	1.50	1.31	3.64	0.93	6.44	31.92
Simuliidae	1.58	0.95	3.55	1.44	6.28	38.20
Amphipod	1.27	1.16	3.30	0.97	5.83	44.03
Hirudinidae	1.00	1.56	2.98	0.96	5.27	49.30
Isopod	1.25	1.23	2.97	1.05	5.26	54.56
Nematocera(pupa)	0.51	0.95	2.95	1.30	5.22	59.78
Oligochaeta	0.86	1.12	2.82	1.34	4.98	64.77
Elmidae	0.67	0.92	2.57	0.63	4.54	69.30
Platyhelminthes	0.06	0.63	2.03	0.57	3.59	72.90
Empididae	0.06	0.67	1.86	0.73	3.28	76.18
Tipulidae	0.21	0.52	1.62	0.92	2.86	79.04
Sisyridae	0.23	0.24	1.35	0.68	2.39	81.42
Ceratopogonidae	0.30	0.15	1.11	0.76	1.97	83.39
Brachycera(pupa)	0.00	0.32	1.04	0.51	1.83	85.22
glassosomatidae	0.06	0.31	0.92	0.65	1.63	86.85
Pyralidae	0.00	0.21	0.82	0.53	1.44	88.29
Hydroptilidae	0.00	0.21	0.70	0.41	1.23	89.53
Tricorythidae	0.12	0.15	0.66	0.52	1.16	90.69

Dates 3 & 4
Average dissimilarity = 59.75

Species	Group 3 Av.Abund	Group 4 Av.Abund	Av.Diss	Diss/SD	Contrib%	Cum.%
Chironamidae	4.06	2.49	7.62	1.09	12.75	12.75
Bivalve	1.23	1.86	4.46	0.84	7.46	20.21
Gastropod	1.95	2.49	4.32	1.26	7.23	27.44
Hydropsychidae	1.31	1.04	3.43	0.87	5.74	33.18
Amphipod	1.16	1.08	3.26	1.18	5.46	38.64
Isopod	1.23	0.46	3.00	1.02	5.03	43.66
Elmidae	0.92	0.34	2.82	0.69	4.72	48.39

Oligochaeta	1.12	1.46	2.73	0.96	4.57	52.96
Platyhelminthes	0.63	0.12	2.71	0.61	4.54	57.50
Hydroptilidae	0.21	0.88	2.20	0.73	3.68	61.18
Simuliidae	0.95	0.00	2.06	1.03	3.45	64.63
Nematocera(pupa)	0.95	0.42	1.95	0.94	3.27	67.89
Hirudinidae	1.56	1.23	1.93	0.88	3.22	71.12
Philopotamidae	0.06	0.66	1.58	0.61	2.64	73.75
Corixidae	0.06	0.31	1.52	0.48	2.54	76.29
Ceratopogonidae	0.15	0.31	1.45	0.48	2.43	78.72
Tipulidae	0.52	0.06	1.21	0.79	2.03	80.74
Empididae	0.67	0.18	1.16	0.82	1.94	82.69
Water mite	0.15	0.50	1.12	0.58	1.87	84.56
Hebridae	0.21	0.15	0.93	0.63	1.56	86.12
Leptoceridae	0.00	0.44	0.92	0.50	1.54	87.66
Brachycera(pupa)	0.32	0.00	0.80	0.52	1.34	89.00
glassosomatidae	0.31	0.06	0.80	0.59	1.33	90.34

Dates 4 & 5
Average dissimilarity = 55.09

Species	Group 4 Av.Abund	Group 5 Av.Abund	Av.Diss	Diss/SD	Contrib%	Cum.%
Chironamidae	2.49	3.86	6.10	1.15	11.07	11.07
Hydropsychidae	1.04	1.83	4.60	0.82	8.35	19.42
Bivalve	1.86	1.47	4.35	1.07	7.90	27.31
Amphipod	1.08	1.48	3.56	1.19	6.47	33.78
Oligochaeta	1.46	2.41	3.39	0.97	6.16	39.94
Hirudinidae	1.23	0.69	3.20	1.03	5.81	45.75
Nematocera(pupa)	0.42	1.29	3.04	1.33	5.52	51.27
Gastropod	2.49	1.89	3.03	1.02	5.50	56.77
Ceratopogonidae	0.31	0.93	2.72	0.63	4.93	61.70
Elmidae	0.34	0.72	2.18	0.69	3.95	65.65
Isopod	0.46	0.52	2.17	0.80	3.94	69.59
Water mite	0.50	0.37	1.92	0.90	3.48	73.07
Hydroptilidae	0.88	0.72	1.79	0.72	3.26	76.33
Philopotamidae	0.66	0.29	1.71	0.60	3.11	79.44
Corixidae	0.31	0.00	1.26	0.42	2.28	81.72
Platyhelminthes	0.12	0.25	1.18	0.57	2.14	83.86
Odontoceridae	0.41	0.41	1.02	0.60	1.85	85.71
Tricorythidae	0.17	0.29	1.01	0.58	1.84	87.55
Leptoceridae	0.44	0.00	0.92	0.51	1.68	89.23
Brachycera(pupa)	0.00	0.40	0.85	0.73	1.54	90.76

Dates 5 & 6
Average dissimilarity = 48.76

Species	Group 5 Av.Abund	Group 6 Av.Abund	Av.Diss	Diss/SD	Contrib%	Cum.%
Chironamidae	3.86	2.10	6.89	1.60	14.13	14.13
Oligochaeta	2.41	1.78	3.68	0.91	7.54	21.67
Nematocera(pupa)	1.29	0.06	3.38	1.71	6.93	28.60
Bivalve	1.47	1.35	3.10	1.20	6.36	34.96
Gastropod	1.89	1.94	3.02	0.97	6.19	41.16
Amphipod	1.48	1.32	2.82	1.08	5.78	46.94
Hydropsychidae	1.83	1.71	2.60	1.02	5.33	52.27
Ceratopogonidae	0.93	0.48	2.34	0.60	4.81	57.08
Water mite	0.37	0.65	2.14	1.03	4.39	61.47
Hirudinidae	0.69	0.56	2.09	0.87	4.28	65.75
Isopod	0.52	0.48	1.83	0.95	3.76	69.51
Hydroptilidae	0.72	0.76	1.55	0.68	3.18	72.69
Elmidae	0.72	0.68	1.48	0.68	3.03	75.72
Platyhelminthes	0.25	0.43	1.20	0.77	2.45	78.17
Odontoceridae	0.41	0.39	1.17	0.65	2.40	80.57
Tricorythidae	0.29	0.32	1.17	0.74	2.40	82.97
Philopotamidae	0.29	0.17	0.95	0.49	1.96	84.93
Tipulidae	0.28	0.12	0.93	0.61	1.91	86.84
Brachycera(pupa)	0.40	0.00	0.90	0.77	1.84	88.68
Baetidae	0.00	0.21	0.75	0.45	1.54	90.22

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